# LONG TERM IMPACTS OF OPEN WATER DISPOSAL

## OF DREDGED MATERIAL

### DRAFT

## PROGRAMMATIC ENVIRONMENTAL IMPACT STATEMENT

### FOR THE

## DISPOSAL OF DREDGED MATERIAL

### IN THE

## LONG ISLAND SOUND REGION

# APPENDIX B

1981

United States Army
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**New England Division**

APPENDIX B

LONG TERM IMPACTS OF OPEN WATER DISPOSAL OF DREDGED MATERIAL

Appendix B

Long Term Impacts of Open Water Disposal of Dredged Material

## Tables

## Figure B-1

I. Introduction

It has only been since the implementation of the National Environ-
mental Policy Act of 1969 (42 U.S.C. Section 4321 et seq.) that the Federal
Government has required its agencies to comprehensively evaluate the conse-
quences or impacts of their actions on the biosphere. The lack of under-
standing of environmental impacts has lead to an increase of research in
the last decade to provide the necessary baseline data to evaluate our
actions.

The study of environmental impacts of dredging and disposal has been
one of the primary tasks of the Dredged Material Research Program (DMRP)
sponsored by the Army Corps of Engineers through the Waterways Experiment
Station at Vicksburg, Mississippi (Saucier et al., 1978). These studies
plus other efforts conducted on both U.S. coasts and in the Great Lakes
region, represent our current knowledge of the effects of dredged sediments
in the aquatic ecosystems of the United States. Although this recent surge
of research has led to a clearer understanding of short-term impacts (less
than 2 years) of open water disposal, impacts over the long-term (2-50
years) are not well understood.

Consideration of the potential for accumulating impacts resulting from
long-term open water disposal of dredged materials has been a major concern
among environmental groups and agencies with an interest in Long Island
Sound (LIS). Long-term cumulative impacts are defined as those "impacts
which result from the incremental impact of the action when added to other
past, present and future actions regardless of what agency (Federal or non-
Federal) or person undertakes such action" (Federal Register November 29,

1978).  Realization of the lack of understanding of such impacts has lead

to consideration of long-term studies.  Current research is directed in

this area as we continue to gather information on our past and current

activities.  This information allows for reasonable analysis of the

consequences of planned disposal activities in the near and distant future

and provides definition for the best approach to gain a better under-

standing of the probability and consequences of long-term effects.  This

information also provides management with an effective approach to minimize

the potential of detrimental long-term effects through siting, timing, and

mitigation techniques.

An example of this kind of effort is the monitoring program recently

sponsored by the New England Division of the U.S. Army Corps of Engi-

neers.  The Disposal Area Monitoring System (DAMOS) program provides a long

term monitoring scheme of disposal activities.  Continual evaluation of

sampling of past and present New England dump sites (including four in Long

Island Sound) over the long term should provide environmental insight into

the following areas of concern:  bathymetric changes, boundary layer turbu-

lence velocities over dump sites (BOLT), suspended sediment monitoring,

sediment chemistry, bioaccumulation of contaminants (mussel watch), benthic

ecology, and fisheries.  The efficiency of capping will also be evalu-

ated.  A more detailed explanation of DAMOS activites is described in Naval

Underwater Systems Center (NUSC, 1979a) and Science Applications, Incorpo-

rated (SAI, 1980).

The scope of this report is to discuss our current knowledge of long-

term impacts in a general way and to apply this knowledge to the potential

disposal operations in Long Island Sound. The complexity of the Long
Island Sound ecosystem, as exemplified by the interrelationships of the
various overlapping trophic levels of the food web, makes predictions
difficult; thus, there are significant limitations in almost all areas of
discussion, e.g., the effects of removal of bottom fauna on foraging fish
populations. The lack of information points to areas in need of further
study.

In general, this appendix discusses the potential physical and
chemical alterations on the Long Island Sound marine ecosystem resulting
from open water disposal of dredged material and its effects on living
systems. This discussion is applied to the candidate dump sites presented
by Dames and Moore (1980) as shown on Figure 1A-2 of Appendix A:

A.  Bridgeport East        $40^{\circ}$ 03.7´ N; $73^{\circ}$ 09.7´ W

B.  Branford               $41^{\circ}$ 10.3´ N; $72^{\circ}$ 47.5´ W

C.  Six Mile Reef          $41^{\circ}$ 10.5´ N; $72^{\circ}$ 30.5´ W

D.  Block Island Sound     $41^{\circ}$ 14.5´ N; $71^{\circ}$ 48.0´ W

E.  Eaton´s Neck East      $40^{\circ}$ 59.6´ N; $73^{\circ}$ 25.0´ W

F.  New Haven              $41^{\circ}$ 08.9´ N; $73^{\circ}$ 53.1´ W

G.  New London             $41^{\circ}$ 16.3´ N; $72^{\circ}$ 04.6´ W


Physical and biological descriptions of these sites are given in Dames and
Moore (1980) Appendix A.

## II. Physical Impacts

### A. Physical Alteration of the Environment:

The open-water disposal of dredged material may result in three major long-term physical changes on the aquatic ecosystem:

1. Change in bottom topography,

2. Alteration of sediment type,

3. Increased potential for physical instability of deposited sediments.

### 1. Change in Bottom Topography

A direct consequence of the increased sedimentation from spoil deposition is an artificial elevation of the bottom topography. Point dumping, i.e. dumping in one spot to confine spoils in a small area, can result in the creation of a mound of considerable size. Such a permanent alteration could cause changes in water circulation patterns, current velocities, water-level fluctuations, temperature, salinity, and ease of navigation if disposal takes place in shallow waters.

The candidate dump sites proposed by Dames and Moore (1980) are all in water at least 20 m deep (Table III-B-1 Dames and Moore, 1980). Dames and Moore (1980) conclude that the sites are deep enough so that these impacts would not be significant. However, Bokuniewicz et al. (1977) point out that an increase in bottom elevation greater than its natural topography can increase the potential for instability of the mounded sediments. This secondary impact will be discussed in Section A3.

## 2. Alteration of Sediment type:

Very often, dredged harbor sediments are physically different from sediments at the dump site. If large enough differences exist in terms of % silt-clay, organic matter, and water, desposition of these sediments may permanently alter the physical nature of the dump site sediments. Coarse bottom sediments (sand and gravel) are generally associated with high energy water currents above them. Deposition of fine sediments may also increase the potential for erosion and thus decrease the physical stability of the mounded sediments. This concept is dealt with in more detail in Section A3.

Benthic communities are known to be associated with specific sediment types (Sanders, 1956; McNulty et al., 1962); therefore, recolonization of deposited sediments after initial burial may result in a change in the community. These changes will be discussed in more detail in Section B1.

The source sediments in this study are the harbors in Long Island Sound. The Environmental Atlas of Long Island Sound Harbor Sediments (CE, 1980) indicates that approximately 70 percent of these harbor sediments have silt-clay fractions greater than 50 percent. Deposition of these fine grain materials would have the greatest impact on sites C and D where the sediments are clean sand with silt-clay contents no greater than 10 percent (Table III-B-1, Dames & Moore, 1980).

In view of the above, the proposed rules for the disposal of dredged or fill material (Federal Register, September 18, 1979) suggest that the source sediments be of a consistency as close as reasonably possible to that of the sediments at the dump site.

## 3. Potential for Resuspension and Transport

The bottom sediments of Long Island Sound are subjected to currents created by the tidal stream, estuarine circulation, waves, and storm induced waves (Bokuniewicz et al., 1977). The long-term stability of deposited dredged material may be violated by these water movements so that resuspension and transport of sediments to areas of greater impact can occur.

Tidal currents superimposed on the estuarine circulation, i.e. the inflow of bottom water from Block Island Sound through the Race, are the dominant source of energy for resuspension and transport of bottom sediment in Long Island Sound (Bokuniewicz et al., 1977). Gordon and Pilbeam (1974) found that 200 mg/cm$^2$ of surface sediments (mostly unconsolidated fluff) are resuspended regularly into the water column near the bottom. This mainly occurs below a depth of 60 feet where wind induced bottom currents do not have a great effect (Bokuniewicz et al., 1977). However, wind-induced currents, especially during storms, may equal or exceed tidal currents at depths less 60 feet. Bokuniewicz et al. (1977) calculated that 3.3 x 10$^4$ yd$^3$ of sediment could be carried southward out of New Haven harbor per tidal cycle during a storm assuming a harbor depth of 1/5 that of the Sound. Thus, wind-induced bottom currents in shallow depths may have considerable potential to resuspend and transport sediment.

Before discussing the potential of resuspension at the proposed dump sites, it is advantageous to explain the interaction of currents and sediment. The following discussion is based on Dames & Moore (1980).

Table 1 (Table IV.F-1, Dames and Moore, 1980) shows the results of studies by Hjulstrom (1935), and Sternberg (1972) relating threshold water velocity at one meter above the bottom to grain size in millimeters. Table 1 shows that fine-grained silts or clays tend to require higher threshold velocities than unconsolidated sandy material. This is due to the increased cohesiveness for smaller grain sizes. Sternberg's results are parallel to, but typically higher than those of Hjulstrom's (Dames & Moore, 1980). The larger grain sizes such as granules and pebbles, also require higher threshold velocities but not to the degree of consolidated clays.

Once suspended in the water column, the size of the particle will affect the minimum velocity required to transport that particle. Table 1 indicates that the larger grain sizes require a higher velocity than silt or clay.

Threshold velocities alone cannot be used to determine threshold velocities for dredged material; other factors tend to complicate the determination. For example, increases in organic content will increase erosional resistance. The established mucous-lined burrows or tubes of polychaete worms and crustacaens are well known to stabilize the surface sediments (Saila, personal communication). On the other hand, the biological activity of recolonized burrowing organisms can increase the water content of bottom materials which in turn, can decrease the erosion resistance of dredged material (Rhoads, 1973). Morton and Miller (1980) found that surface roughness also can decrease resistance. "At present, no detailed data is available to quantify the change in erosional threshold velocities of different materials as a function of the extent of consolidation or burrowing activity" (Dames & Moore, 1980).

TABLE B-1[1]

EROSION AND TRANSPORT THRESHOLDS FOR INITIATION OF GRAIN MOVEMENT

| Sediment Type | Minimum Water Velocity Required[2] (1 m above the bottom) | | | Minimum Water Velocity Required[3] (1 m above the bottom) |
|---|---|---|---|---|
| | Particle Diameter (mm) | Erosion (cm/sec) | Transport (cm/sec) | Erosion (cm/sec) |
| Consolidated clay | 0.002-0.004 | 100 | 0.1 | -- |
| Silt | 0.004-.04 | 27 | 0.1 | 29 |
| Sand | .04-2.0 | 20 | 0.5 | 29 |
| Granule | 2.0-4.0 | 40 | 17 | 50 |
| Pebble | 4.0-5.0 | 60 | 30 | 70 |

[1] Table IV - F-1 (Dames and Moore, 1980)

[2] Hjulstrom (1935)

[3] Sternberg (1972)

In the absence of an applicable erosional threshold velocity for dredged sediment, Dames & Moore (1980) interpolated a value based on Bokuniewicz and Gordon's (1977) comparison of two Long Island Sound dump sites. Bokuniewicz and Gordon (1977) noted that cohesive silty dredged material at the New Haven dump site did not erode with a bottom current velocity in the range of 25-30 cm/sec while erosion and resuspension of similar materials did occur at the Cornfield Shoal Site which has a maximum bottom tidal velocity range of 40-55 cm/sec. Thus, Dames & Moore (1980) concluded that an effective threshold velocity for resuspension of silty dredged materials is in the range of 35-40 cm/sec.

In view of the lack of on site data, caution must be exercized in applying this velocity range and the calculated tidal and wave current velocities for the candidate sites. In the future, in situ measurements of those velocities are proposed in the Boundary Layer Turbulence (BOLT) study as part of the DAMOS Program.

a.  Tidal Currents

Tidal currents are the dominant source of energy for the resuspension and transport of sediments (Bokuniewicz et al., 1977). Dames and Moore (1980) extrapolated plausible tidal bottom currents for the candidate sites A, B, C and D based on available tidal current measurements from Sites E, F and a location in eastern Long Island Sound (Table 2). Comparison of these values with the interpolated erosional threshold velocity 35-40 cm/sec

| Site | Significant wave height[2] (meters) | Significant wave period[2] (seconds) | Wave-induced Bottom Currents for specified height/period (cm/sec) | Tidal Current and net direction[3] (cm/sec) | Sedimentation Rate $(kg/m^2/yr)$ | Potential Confinement of Sediment |
|---|---|---|---|---|---|---|
| A  Bridgeport | 0.8-0.9 | 3.4-3.6 | 0 | 20-25 W+SW | 0.5-0.8 | good |
| B  Branford | 1.2-1.3 | 4.3-4.5 | 3-5 | 25-30 E+ENE | 0.8 | good |
| C  Six Mile Reef | 1.2-1.3 | 4.3-4.5 | 1-2 | 50-55 E | 0 | poor |
| D  Block Island Sound | 2.7-3.1 | 7.0-9.0 | 17-40 | 35-45 E+SE | 0 | fair |
| E  Eaton's Neck | 0.6-0.7 | 3.0-3.2 | 0 | 28-33 W+SW | 0.3-0.4 | good |
| F  New Haven | 1.2-1.3 | 4.3-4.5 | 2-3 | 27-31 W | 0.7-0.8 | good |
| G  New London | 1.2-1.4 | 4.4-1.7 | 4-7 | 40-45 E+NE | 0 | fair |

(1)  Excerpted from Table III-B-1 (Dames and Moore, 1980)
(2)  Calculated from longest effective fetch, during 20-22 knot winter storm (Dames and Moore, 1980)
(3)  Extrapolated from available data from Eaton's Neck, New Haven and Eastern Long Island Sound (Dames and Moore, 1980)

gives an indication of potential for long-term erosion at each candidate site. It appears that Site C in central LIS has the highest tidal velocity and thus the greatest potential for resuspension. The sedimentation rate (Table 2) and coarseness of the bottom are consistent with this evaluation (Dames and Moore, 1980). The next highest maximum bottom tidal currents are found at site G (40-45 cm/sec). "This is apparently a borderline case suggesting a potential for occasional erosion and transport" (Dames and Moore, 1980). The same may be true for site D which exhibited calculated tidal current velocities of 35-45 cm/sec (Table 2). Again, the lack of long term sediment accumulation at these sites (Table 2) indicates that erosion of fine sediments is probable because sedimentation is less likely to occur. Long term monitoring of bottom currents would be needed for verification.

Sedimentation does occur at Sites A, B, E and F with maximum tidal velocities ranging from 2-30 cm/sec. Such sites would therefore be more suitable for confinement of contaminated material whereas sites C, G and D would favor coarse or less contaminated material.


b. Storm Induced Wave Currents

Bokuniewicz et al. (1977) found that at depths greater than 60 feet, bottom sediment were not likely to be eroded by wind-induced currents. Even during a 40 knot storm, Bokuniewicz et al. (1977) noted that only about 2 mm of sediment was resuspended at a location one mile south of Site F (New Haven). However, the authors further noted that, as the deposition of dredge material is higher than the surrounding sea floor, it is exposed

to higher current velocities. These areas are more susceptible to erosion during storms.

Dames & Moore have calculated the wave-induced bottom current velocities of the candidate sites based on the typical 20-22 knot winter storm. The siginficant wave height and period (Table 2) used in the calculations were taken from data in Block Island Sound during the winter. Calculated bottom currents were below Dames and Moore's threshold value (35-40 cm/sec) in all cases except Site D (17-40 cm/sec, Table 2), where it could be exceeded indicating that Site D has wave-induced erosion potential.

Long-term monitoring of disposal mounds will give an indication of their stability during storms of higher magnitude. The Brenton Reef site in Rhode Island has been well studied after disposal discontinued in late 1970. Diving surveys in 1974 and 1977 revealed ripple marks and winnowing of fine material at the mound summit which indicates reworking by bottom currents (Chase 1974; Chase, personal communication, 1980). It must be pointed out that Brenton Reef is in a open, unprotected area and therefore would be more subject to erosion. Most of the LIS sites are protected by Long Island. The intermediate-term (1-3 years) monitoring that has been done at LIS dump sites has indicated varying results. Cobb et al. (1977) noted that after disposal was terminated in 1974 at the Eaton's Neck Dump Site, near site E, surveys in 1974 and 1976 indicated no appreciable bathymetric changes beyond the detection limits of the survey techniques. Similarly Bokuniewicz et al. (1977) reported that repeated bathymetric surveys of a deposit of dredged material at site F showed that, after the

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initial self-consolidating of the mound, no significant changes in pile

configuration occurred over a 3 year period. This includes the passage of

Hurricane Belle which had wind speeds up to 81 knots. The lack of fetch

combined with the short duration of such high velocity storms limited the

intensity of the bottom currents. However, a late 1979 survey of a mound

0.3 miles south this site showed about 10,000 cubic meters or 12 percent of

silty capping material had been removed from the southern spoil mound

summit probably during the passage of Hurricane David in September 1979

(Morton & Miller, 1980). The original New Haven dump site and a northern

sand capped mound were not affected. Morton & Miller concluded that the

roughness of the spoil mound (i.e., morphology) under storm conditions is

more significant than the depth of the mound. The authors further con-

cluded that smoothing of the surface and sand capping would have minimized

the erosion. Although no detectable erosion at the New London Site

occurred during the 1975-1976 winter storm season and Hurricane Belle (U.S.

Navy, 1979), recent diving surveys (Stewart, 1978) have indicated that

surface erosion was evident. Winnowing of fine surface material and

extensive shell surface lag material were observed. However, the amount of

material transported was not significant (U.S. Navy, 1979). It is apparent

that the extent of erosion of mounded sediments relative to wave induced

bottom currents still needs further study. Long-term monitoring of sedi-

ment movements under a variety of wave conditions is currently underway as

part of the DAMOS Program. In view of the potential for erosion and

transport of capping material, the efficacy of capping and its long-term

stability also will be evaluated on site under the DAMOS program.

B. Long-term Biological Impact from Physical Alterations

1. Biological Impacts from Habitat Alteration

a. Changes in sediment type

Disposal operations will bury most benthic organisms at the dump

site. Although infauna, especially the deeper forms, have a greater chance

of survival (Saila et al., 1972; Maurer et al., 1978), the short term

result is the general loss of benthic productivity available as food for

higher trophic levels such as demersal fish. Recolonization may begin

within days after disposal and may take anywhere from 1.5 (U.S. Navy, 1979)

to 11 years (calculated by Saila, 1973) for complete recovery to occur,

provided that subsequent dumping does not occur. Complete recovery may be

severely hindered if the disposal sediments differ markedly from the

natural site sediments in terms of grain size, organic content, water

content and chemical constituents.

Benthic communities are well known to be associated with sediment

grain size (Thorson, 1957; Sanders, 1956, 1958; McNulty et al., 1962;

Rhoads et al., 1974). In general, high silt sediments contain a large

proportion of deposit feeders whereas sandy sediments have a large

proportion of filter feeders. Deposit feeders are generally found in low

energy regimes where finer particles settle in the sediment and increase

the organic content of the sediment. Such species are more tolerant to the

low dissolved oxygen associated with these sediments. Filter feeders, on

the other hand, feed on the organic particles which remain suspended in water column under a high energy regime. The water current also supplies the high dissolved oxygen required by many of these feeding types. Johnson (1971) found that these general requirements allow mud species to more easily invade sandy substrates than sandy species are able to invade mud. The Ocean Disposal Conference (1971) and Maurer et al. (1978) suggested that physical impacts due to sediment alterations are minimized when sand is placed on a sandy bottom and maximized when mud is placed on a sand bottom. The introduction of high silty-clay sediment associated with most harbors would hinder recolonization of sandy or silty-sand biota found at sites C and D. The number of species and individuals may be altered at these sites. Thus, disposal of mud on sandy substrates is an artificially or man-induced selection mechanism which favors deposit-feeding or opportunistic species. The effect of this change on local foraging benthic feeders such as demersal fish, whether beneficial or adverse is unknown.

b. Increased Periodic Sedimentation and Community Instability

Following disposal of dredge material at the New Haven dump site in 1974, Rhoads et al. (1978) studied recolonization of the disposal mound for a 26 month period. A distinctive succession of colonization was seen occur much in the same pattern as McCall (1977) reported in his tray experiments about 1.5 km northeast of the dump site. Early short-lived colonizers such as the polychaetes Streblospio benedicti, Capitella capitata, the amphipod Ampelisca abdita, and the bivalves Mulinia lateralis, were first seen. Their presence set the stage for colonization by larger longer-lived

benthic invertebrates such as bivalves <u>Nucula annulata</u> and <u>Tellina</u> <u>agilis</u>. Eventually, a climax community composed of even larger and longer-lived species such as polychaete <u>Nepthys incisa</u> and the molluscs, <u>Ensis</u> <u>directus</u> and <u>Nassarius trivittatus</u>, gradually replaced the former groups. The higher turnover rates associated with the first two groups actually enhanced the productivity at the dump site six times that of a nearby reference site for a period of 2-3 months, 200 days following disposal. Continued annual disposal will likely cause repetition of this same sequence with productivity varying from a loss during disposal to enhanced productivity for a 2 to 3 month period, 200 days later. The overall effect is long-term instability in the benthic community productivity which may or may not have an impact on foraging predators. More study in this area is needed. The effects of continued unstable benthic populations on higher trophic levels has never been studied. However, the overall impact on LIS from changes and instability in community structure may be characterized by the fact that the total area of the seven candidate sites (A-G) (Dames and Moore, 1980) of 8 square nautical miles is less than 1 percent of the total area of Long Island Sound. If impacts are confined to the disposal area, the quantitative effect may be small.

## 2. Biological Impacts from Sediment Resuspension

The impacts of suspended sediments on biological systems are generally considered short-term (Stern and Stickle, 1977) because of the short suspension time of most particulates. However, the deposition of fine grained sediment on high energy site (such as Site C) may cause long term

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intermittent resuspension of fine particulates depending on the magnitude of the tidal currents.

Peddicord et al. (1978) exposed a number of benthic and pelagic species to uncontaminated and contaminated sediment suspensions of varying concentrations. In general, all species tested were able to survive exposure to much higher concentrations of relatively uncontaminated sediment for longer periods of time than created in the water column by typical dredging and disposal operations. Fluid mud concentrations, similar to those formed during hydraulic dredging operations, for periods of 21 days or more only slightly increased the mortality in the shrimp Crangon nigromaculata and, the mussel, Mytilus californicus and in juvenile striped bass Morone saxtillis. Similar exposures with contaminated sediments did cause significant mortality in the mussels, Mytilus edulis, and M. californicus, the tunicate Ascidia ceratoda, the crab Cancer magister, the shrimp Crangon nigromaculata and the juvenile striped bass. Exposures to the juvenile lobster Homarus americanus did not show significant mortality. Such concentrations or exposure times would not likely occur during tidal or wave-induced resuspension and therefore should not be a problem if the sediments are relatively uncontaminated.

However, if the sediments are contaminated with potentially hazardous substances resuspension could increase their release into the water column and availability to organisms.

III. Chemical Impacts

Approximately 90 percent of the natural and anthropogenic contaminants carried down the rivers, which empty into central and western Long Island Sound, accumulate in the sediments (Fitzgerald et al., 1974). Natural sedimentation in harbors accounts for a good portion of these constitutents which are associated with the suspended load. The maintenance of harbors for the passage of vessel traffic necessitates the periodic dredging and displacement of sediment in selected areas.

The mechanics of this activity may induce release of these foreign substances into the water column or interstitial water during the operation at the dredging and disposal sites (Burks and Engler, 1978). These effects are temporary and are generally restricted to the dredging and disposal areas. A major concern is the long-term stabilization of these contaminants at the disposal site.

A. Loading of Contaminants into Long Island Sound Sediments.

1. Long Island Sound Harbor Sediments.

Bulk sediment analyses of the Long Island Sound harbors indicate that Western Long Island Sound, west of Stratford Shoals (CE, 1980) has the highest percentages of silts and concentrations of contaminants (i.e., oil and grease, volatile solids, heavy metals, chlorinated hydrocarbons) than the remainder of the Sound. Harbor sediments in the Eastern Sound, east of Mattitick Sill, contain a lesser amount of silt and contaminants. The Central Sound harbors are generally intermediate although some contaminants approach either extreme.

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West Chester, Black Rock, Bridgeport, Stamford, and Greenwich harbors in the Western Sound have the highest incidence of contaminants which occur at levels considered by the Interim Plan on disposal of dredged material in Long Island Sound (NERBC, 1980a) to have the greatest potential for impacting the environment (CE, 1980). The incidence and degree of con- taminant decreases toward the eastern part of the Sound into Block Island Sound.

2. Long Island Sound Basin Sediments

The "background" levels of 13 trace metals in the Long Island Sound surface sediments have been reported by Greig et al. (1977). In general, the authors found that all metal concentrations except four were highest in the Western Sound, intermediate in the Central Sound and lowest in the Eastern Sound. Metal levels in these surface sediments were generally lower than those of the harbor sediments, as listed in the Environmental Atlas of Long Island Sound Harbor Sediments (CE, 1980). Therefore, intro- duction of contaminants by disposal of dredged sediments from most of the Western Basin harbors and some of the Central Basin harbors would permanently elevate the "backround" concentrations in the disposal site sediments. Dumping on past disposal sites such as F and G would probably reduce the severity of this impact because of already elevated levels.

The long term impacts associated with increased sediment contaminant levels are the potential release of these substances into the water column and the lethal and sublethal physiological effects of the contaminants to benthic and pelagic organisms. Recent studies have shown that the degree

of these impacts is not directly related to contaminant levels indicated by bulk sediment analysis alone, but is also related to the chemical form of the contaminants and the chemical environment in which they exist (Lee and Plumb, 1974, Chen et al., 1976, Brannon et al., 1976, Khalid et al., 1977, Neff et al., 1978; Brannon et al., 1978; Brannon, 1978).

## B.  Mobility and Availability of Contaminants

Contaminants in sediment generally exist in a variety of chemical forms depending primarily upon the nature of the sediment (Brannon, 1978).  Some of these forms exist as tightly bound complexes which render the contaminants immobile and thus innocuous to the environment.  Other forms are more easily mobilized by alteration of sediment chemistry (pH and reduction-oxidation state) which, in turn, can affect the water chemistry.

### 1.  Heavy Metals

#### a.  Geochemical Forms

Heavy metals, in particular, occur in a variety of geochemical forms with a wide range of mobility.  The forms in Table 3, based on an extraction procedure developed by Engler et al. (1974), are in order of decreasing mobility.  "Interstitial water" and "Ion-exchangeable" phases are the most mobile and thus are easily released into the water column. Their levels are established by an equilibrium between the sediments and the water column.  Changes in salinity, cation concentration, pH, and oxygen concentrations can disrupt this equilibrium until a new equilibrium is established.  "Organics & Sulfides", and "Reducible" phases are poten-

tially mobile. Changes in pH or reduction-oxidation states of the sediments will alter their respective equilibriums. "Easily Reducible" is more mobile than "Moderately Reducible." The crystalline oxides of the "Moderately Reducible" phase are generally more binding to metals than the amorphous oxides of the "Easily Reducible" phase and thus have more of a tendancy to immobilize metals. Metals in the "Residual phase" are tightly bound in the crystalline lattice of the sediment minerals and are essentially inert and biologically unavailable. The majority of the naturally occurring metals are found in this phase.

Table B-3
Geochemical Forms of Heavy Metals in Marine Sediments
based on Extraction Procedure of Engler et al. (1974)

| Geochemical Phase* | Chemical Species |
|---|---|
| Interstital Water | Free cations, chlorides, elemental forms |
| Ion-exchangeable | Cations sorbed to negatively charged exchanged sites on clays, Fe & Mn oxides, organic particles. |
| Easily Reducible | Managanous oxides, some amorphous iron oxides, carbonates, insoluble hydroxides |
| Organic & Sulfides | Humic and Fulvic acids, sulfides, bisulfides |
| Moderately Reducible | Hydrous crystalline iron oxides |
| Residual | Crystalline lattice of minerals |

*Phases are in order of decreasing mobility and bioavailability.

Although the distribution of metals in these various geochemicals phases can vary from sediment to sediment, the major fraction of most metals is usually associated with the "Organic & Sulfide", "Moderately Reducible" and the "Residual" phases in marine sediments (Brannon et al., 1976; Serne, 1977). Brannon et al. (1976) have analyzed the metal geochemical form distribution of the Bridgeport Harbor sediments which may be an example of other Long Island Sound Harbors (Table 4). In all cases except zinc (Zn), 95 percent of the total bulk chemical levels are found in the less mobile "Organic & Sulfide", "Moderately Reducible" and "Residual" phases. Eighty-five percent of the Zn is in the latter phases.

Bulk chemical analysis of sediment involves the acid digestion of the sediment; therefore, the entire elemental content of that sediment including the metals associated with the "Residual" phase is measured. The total concentration of metals measured by bulk analysis would be higher than those available in the marine environment since a major fraction of the metals may remain immobile and unavailable under an appropriate chemical environment (see below).

Table B-4
Metal distribution (%) among various
geochemical phases of Bridgeport Connecticut
Harbor sediment*

| | Fe | Mn | Cu | Zn | Ni | Cd | As |
|---|---|---|---|---|---|---|---|
| Interstitial | 0.0007 | 0.0005 | 0.000006 | 0.10 | 0.00006 | 0.001 | 0.009 |
| Exchangeable | 0.18 | 0.25 | - | 0.33 | 0.024 | 0.00 | 0.00 |
| Easily Reducible | 2.07 | 2.30 | 0.008 | 13.73 | 1.64 | 4.32 | 1.66 |
| Organic & Sulfide | 8.84 | 30.81 | 92.00 | 56.48 | 25.35 | 95.68 | 0.00 |
| Moderately Reducable | 35.01 | 3.75 | 0.46 | 22.11 | 0.43 | 0.00 | 70.44 |
| Residual | 53.90 | 62.89 | 7.60 | 7.24 | 72.56 | 0.00 | 27.90 |
| Total Concentration ppm (acid digestion) | 43600. | 531.5 | 11.17 | .967. | 203.5 | 17.60 | 6.90 |

*from Brannon et al. (1976).

b.  Chemical Environment.

As mentioned above, metals in the residual phase are tightly bound to the crystalline lattice of minerals so that only strong acid digestion can release them into soluble forms.  However, metals in the "Organic & Sulfide" or "Reducible" phases may be transferred to more soluble forms with changes in the oxidation state or pH (Gambrell et al., 1977).

A reduced or anaerobic sediment generally favors the immobilization of most heavy metals, with the exception of iron (Fe) and maganese (Mn), in insoluble organic and sulfide metal complexes which are generally unavailable to water column organisms.  Gambrell et al., (1978) in his recent review of current literature, found this to be true for mercury (Hg), lead (Pb), cadmium (Cd), zinc (Zn), copper (Cu), arsenic (As), nickel (Ni), and chromium (Cr).  In general, lowering of the pH or increasing the oxidation state of reduced sediments may change these forms into more soluble phases (Gambrell et al., 1977).  For instance, a significant proportion of the total cadmium content was transformed into soluble and exchangeable forms under pH values of 5.0 and 6.5 when reduced sediments were slowly oxidized (Gambrell et al. 1977).  However, if sufficient amounts of Fe and Mn occur in the sediments, oxidation of such sediments will immediately lead to the precipitation of hydrated Fe and Mn oxides which scavenge most of the other trace metals that might be released making the latter unavailable (Chen et al., 1976).  Extreme reductions in pH may lead to significant mass release of heavy metals in soluble form (Brannon et al., 1978).  Such conditions could occur under upland disposal conditions when oxidation of sediments will lead to the formation of sulfuric acid which lowers the pH of the

surface sediments. Under these conditions, drainage can leach high concentrations of these metals out of the sediments. This marked change in pH is unlikely to occur in open water dipsosal where the pH of the water and sediment is buffered by the natural carbonate content of marine sediments. The potential for oxidation of deposited sediment is increased if dumped in such high energy sites as C, D and G. Thus "problem" dredged materials with moderate or high levels of contaminants should be dumped at low energy areas such as A, B or F and capping might be considered to maintain integrity of the reducing environment.


c. Prediction of Potential Release of Contaminants into the Water Column.

The standard elutriate test has been jointly devised by the Corps of Engineers and EPA to predict the potential of contaminants release into the water column (EPA/COE, 1977). The sediment is mixed with sea water usually collected at the dump site. Brannon et al. (1976) compared elutriate test values of contaminants with those of the separate extractions in each geochemical phase. Their results indicated that the standard elutriate metal concentrations were correlated with almost 78 percent of the levels in the "Interstitial", "Exchangeable" and "Easily Reducable" phases. Although Zn and Ni from the "Organic & Sulfide" phase were released in a few instances, the authors concluded that significant amounts of metals associated with this phase would not be mobilized based on the harshness of the extraction procedure. Brannon et al. (1978) evaluated the elutriate test as an indicator of long-term release of contaminants. Metal and nutrient concentrations from long-term leaching experiments were compared

with elutriate concentrations. Long-term leaching experiments from a wide variety of sediments under conditions likely to prevail at the disposal sites, indicated consistent net releases of the total organic carbon, phosphate, and Zn. Very little mass release of As, Cd, Hg, and Pb were observed regardless of the composition of the sediments. Comparison of these values with the standard elutriate test indicated the latter together with the interstital water analyses were useful in prediction of potential long-term releases of As, Cd, Hg and Pb. However, the experiments of Brannon et al. (1978) only lasted for a period of 8 months. It is not known whether these tests can predict long-term releases that will occur over a 20 to 50 year period. Over such a period, sediment resuspension and/or sediment reworking by recolonizing benthic organisms to a depth of 10-20 cm may alter the sediment oxidation state so that unavailable metals are transferred into more soluble forms. Such transformations could increase the interstital water concentrations of metals which could eventually diffuse into the overlying water column causing localized increases. Although diffusion is normally a slow process, it may be hastened by sediment resuspension. Thus, the elutriate test and interstitial water analyses should be coupled with analyses of geochemical forms of the sediments to indicate potential long-term release. A systematic study of the transformation of the metal-geochemical complexes with corresponding changes in pH and reduction-oxidation states of Long Island Sound harbor sediments could shed light on the long-term release potential of heavy metals.

## 2. Organic Compounds

### a. Oil and Grease

Oil and grease is a general term used to describe both naturally produced fats, oils and waxes as well as an extremely broad array of petroleum derivitives and synthetic organic chemicals, most of which are insoluble in water. These include the numerous simple and polynuclear aromatic compounds some of which may have toxic and/or carcinogenic or mutagenic properties (e.g. Benzene, toluene, benzapyrene). Although many synthetic and straight-chained petroleum hydrocarbons can be decomposed by sunlight or by aerobic microbiological activity (Shelton and Hunter, 1975), many complex petroleum hydrocarbons are persistent for many years especially in an anaerobic environment (Blumer and Sass, 1972).

Field and laboratory investigations have shown that relatively small amounts of hydrocarbons are released into the water column above contaminated sediments based on the general insolubility of these fat soluble substances in water and the absoptive capacity of sediments (Chen et al., 1976; Engineering Science, 1977; DiSalvo et al., 1977). Fine sediments especially have a high adsorptive capacity because smaller particles have a greater surface area per unit volume and are capable of adsorbing large quantities of oil. Although some petroleum aromatics are known to be slightly soluble, experiments have shown little or no release of soluble hydrocarbons residues to the water column (Chen et al., 1976; DiSalvo et al., 1977). Resuspending sediments containing aromatic compounds could cause deleterious effects upon aquatic organisms (Burks & Engler, 1978).

The current state of the art has not indicated significant release of heavy oil fractions in the water column (Burks & Engler, 1978). DiSalvo et al. (1977) found that the elutriate tests were inadequate to predict water column release due to the adsorption of oil and grease on the walls of the test vessels.

b. Chlorinated Hydrocarbons

Polychlorinated biphenyls (PCB) and chlorinated hydrocarbon pesticides such as DDT are among the most toxic and persistent of contaminants and may be found in dredged sediment. Their presence is the result of man's use of these compounds. Current Federal regulations have severely restricted the release of these contaminants into the environment for the past several years. However, harbor sediments containing these substances from past release may persist for many years (Burks and Engler, 1978).

As with petroleum hydrocarbons, chlorinated hydrocarbons are fat soluble, hydrophobic and are generally tightly bound to fine sediments and organic particulates (Chen et al., 1976). Yet under certain conditions these substances can be released into the water column (Fulk et al., 1975; Lee et al., 1978; Wright, 1978).

Fulk et al., has shown that the sediment to seawater ratio can influence the release of PCB's in the water column. Release experiments with varying sediment to liquid ratios indicated that sediment to liquid ratios exceeding 20 percent (one part sediment to five parts water) were necessary before any release of chlorinated hydrocarbons occurred. Since sediment to liquid ratios exceeding 20 percent would not occur during most

mechanical dredging operations, it was concluded that release during these operations would be minimal. The same principle would apply after disposal when resuspension of these solids can occur.

The total oil and grease content has also been shown to influence water column levels of chlorinated hydrocarbons. Jones and Lee (1978), using the elutriate test found that when the sediment suspension was 5 percent, there was an inverse relationship between PCB release and the amount of oil and grease contained in the sediment. This is based on the solubility of these compounds in oil and grease and the insolubility of these fractions in water. However, when the sediment-liquid ratio was increased to 20 percent, this relationship did not hold. Fulk et al. (1975) in their leaching study found that there was a direct correlation between PCB´s remaining in the water column and the oil and grease content increase sediment sorbtion of chlorinated hydrocarbons under low suspended solids or magnify release when suspended solids are high. The analyses of Fulk et al. (1975) indicated that the oil concentration in the suspended sediments, when present, was about 20 times more important than the quantity of suspended material in affecting the PCB concentration in suspension.

However, settling studies indicated the chlorinated hydrocarbon concentration associated with suspended solids returned to relatively normal levels after 24 hours of settling under quiescent conditions. Thus, water column elevations would occur only with continuous resuspension.

At least four Connecticut harbors, are known to have concentrations of chlorinated hydrocarbons, are Bridgeport, Black Rock, Greenwich and Thames

River. They also have moderate to high oil and grease levels (CE, 1980).
Therefore, it is suggested that these sediments be deposited in low energy
sites such as A, B and F where sedimentation occurs and resuspension
potential is minimal. Although, the sediments resuspended may be well
below 20 percent, it is uncertain how much the oil and grease content will
contribute to polychlorinated hydrocarbons availability especially if
resuspension is frequent as in a high energy dump site such as Site C.
Jones and Lee (1978) have concluded that the elutriate test is a valuable
tool for prediction of potential release of these substances.

## 3. Nutrients

Nutrients such as Kjeldahl nitrogen, ammonium nitrogen, nitrates,
nitrites and ortho-phosphates are generally not considered to have long-
term impacts (Gambrell et al, 1978). Substances such as these are
generally biodegradable but may cause short-term algae blooms and/or
temporarily depressed dissolved oxygen levels in unmixed areas such as
Western Long Island Sound.

## 4. Summary of Water Quality Impacts

Long-term water quality impacts from disposal of dredged material
involves the potential release of persistent contaminants such as heavy
metals, chlorinated hydrocarbons, and petroleum residues into the water
column. Heavy metals in dredged material are bound in various geochemical
forms. Analysis of these forms has revealed that a major portion of the
total metal content is biologically unavailable and would probably remain

so as long as the disposal mound remains disturbed. The same is true for persistent hydrocarbon compounds which have a tendency to be bound to organic matter or fine sediments. Release of such contaminants by transformation into soluble or potentially bioavailable forms may be increased by sediment reworking of benthic organisms and/or by disposal on high energy dump sites where bottom curents (tidal or wind-induced) can resuspend the sediment. Blanketing the contaminated dredged material with a cleaner cap material could block any potential sediment-water interface activities and allow the contaminant to remain sequestered in the mound. The efficiency of capping over the long-term will be monitored and evaluated under the DAMOS program.

## C.  Chemical-Biological Interactive Effects

The biological uptake of persistent chemical contaminants is the major long-term concern of disposal of contaminated dredged material. Aquatic organisms are in a state of dynamic flux with their chemical environment in which soluble substances are actively or passively accumulated from and released to water and sediments. Uptake of available contaminants occurs in two ways: (1) absorption through gill membranes and/or skin, (2) direct ingestion. Contaminants may be either assimilated or eliminated through excretion, defecation or simple diffusion. Depending on the toxicity and persistence of the contaminant incorporated into the organisms tissues, it may be hazardous to the organism's health, the health of the population, or the health of any consumers and their populations. Although the latter has been demonstrated to be related to other forms of disposal

such as acid waste, it has not been shown to be directly caused by the disposal of dredged material.

## 1. Uptake from Water

The uptake and assimilation of soluble toxicants from the water column has been well established in numerous studies (EPA, 1976). In general, the concentration factor, i.e., the concentration of contaminants by an organism divided by the ambient water concentration, will increase with increasing duration of exposure, exposure concentration, temperature and with decreasing salinity (Neff et al., 1978). Thus, under the right conditions, organisms have been known to accumulate contaminants many times the ambient concentrations (Waldichuk, 1974).

The highly soluble methyl mercuric chloride is of particular concern. Liquid methyl mercuric chloride was dumped in large quantities into Minimata Bay, Japan. Accumlation of this highly toxic substance in fish was the source of the infamous Minimata Disease (Fujiki, 1973). Organic contaminants also show a similar pattern. Scura and Theilacker (1977) found that accumulation of PCB's in anchovy larvae from the water column was dependent on the PCB concentration in sea water.

## 2. Uptake from Food

Contaminant-laden food has also been shown to be a source of uptake of aquatic organisms. Instances have been shown that food is a source of tissue contaminants for all types of feeding mechanisms: Grazers (Young et al., 1975); filter feeders (Kerfoot & Jacobs, 1974, Harrison et al., 1976);

deposit-feeders (Luoma and Jenne, 1975; Neff et al., 1978); and predators (Jarvenin et al., 1977).

## 3. Uptake from Sediments

Uptake is also known to occur directly from sediments. In general, Neff et al., (1978) found that uptake varied with the metals involved, geochemical forms of the metal, sediments, species, season, temperature and salinity. Accumulation was not related to bulk chemical levels in the sediments. One might expect that recolonizing benthic organisms which are intimately associated with disposed sediments may the most susceptable to accumulate contaminants. Lowman et al. (1971), Wolfe and Rice (1972) and Schubel (1972) have suggested that sediment bound heavy metals may be an important source of heavy metals for benthic infauna and deposit feeding organisms. However, Neff et al. (1978) question the direct contribution of the sediment based on the difficulty to separate sediment uptake from water uptake when both sediment and water must be present in the same aquarium during experimentation. Biological activity in sediment can cause the release of metals into the water of a static aquarium which, in turn, can be accumulated by the test animals. Direct accumulation from the sediments could not be distinguished from accumulation from the static water column. This would not occur in situ because sediment contaminants that were released into the water column would be diluted and carried away by bottom currents. Thus benthic organisms in the immediate area would not be significantly exposed. Neff et al. also maintain that whole body analyses used in many studies may have greatly overestimated body burdens. For some

elements, more than 90 percent of the body burden of some test organisms
can be associated with gut content and surface contaminants (Elwood et al.,
1976, cited by Neff et al., 1978). Some studies have compared the relative
contributions of uptake from sediment and water. Ueda et al. (1976, 1977,
cited by Neff et al., 1978) found that uptake of Cd, Co, Cr, Nb, Ru, Cs
from water ranged from 6-1,000 times more efficient than from sediment in
the polychaete, Nereis japonica. Similar results were seen in the study of
Amiard-Triquet (1975), also cited by Neff et al., (1978) concerning the
accumulation of Co and Cs in four deposit-feeders.

Based on the relative infrequency and marginal uptake exhibited in the
literature as well as in their own study, Neff et al. suggested that metal
uptake from sediment was of doubtful ecological significance. Certainly,
more work is needed for substantiation of this statement.

4. Effects on Aquatic Organisms

Once an organism accumulates and assimilates contaminants, the effect
on that organism may be acute or chronic. If the concentration in the
tissue is high enough, severe physiological impairment of the organism may
result in death. The toxicant may act alone or together with other contam-
inants or detrimental physical effects such as dissolved oxygen depletion
to cause the same final result. When such synergistic effects occur, the
toxicants may not have to be in lethal doses to cause death. The bioassay
test has been developed to estimate such effects. The EPA and Corps of
Engineers now utilize bioassay tests to give an indication of short-term
toxicity of specific harbor sediments on the representative sensitive

organisms. However, this does not give an indication of the chronic

effects of sublethal contaminant levels in organisms which are of a major

long-term concern. Although many studies have indicated reproductive,

developmental growth impairment, deviation in behavior, as well as

mutagenic and carcinogenic effects from sublethal doses (EPA, 1976), the

complexity of environment and the high variabilty among species make

worthwhile prediction of such effects on an entire community impossible.

Bioaccumulation tests are used to determine sublethal levels of accumulated

contaminants in tissue in field and laboratory situations. Although this

is a step in the right direction, without a reasonable grasp of sublethal

effects in relation to these levels, its use as a predictor of long term

impacts is limited.

These tests used in evaluating disposal options are discretionary with

the permitting agencies. However, large Federal projects currently do

involve bioassay and bioaccumulation testing as a matter of practice.

Another potential chronic impact as a result of the accumulation of

toxicants is magnification of these substances up the food chain. Bio-

magnification is the process by which tissue concentrations of accumulated

chemical residues increase by multiples of whole numbers (2x, 5x, 10x) when

these materials passed up the food chain through two or more trophic levels

(Macek et al., 1979). According to Macek et al., (1979), this process, as

it is defined, has only been truly demonstrated in DDT (Jarvenin et al.,

1977). Although purported to occur for mercury and PCB´s, Conner et al.

(1979) indicates that true biomagnification has not been demonstrated.

Transfer of contaminants to a higher trophic level in both cases resulted

from ingestion of organisms which had accumulated large amounts of contami-
nants from the water column (Britt and Hushon, 1976; Kornreid et al., 1976,
as cited by Conner et al., 1979). These studies have not shown these
substances to be magnified two or more times over two or more trophic
levels. This is not to say that biomagnification does not occur with these
and other substances or that it will not be demonstrated in the future.
Unfortunately, information on these matters usually occurs after the
fact. The Corps of Engineers does monitor disposal sites under the DAMOS
program to anticipate such problems.

D.  Field Investigations

The above discussion, based mostly on laboratory experiments, gives an
indication of the potential biological problems associated with a contami-
nated environment. However, the applicability of laboratory data extra-
polated to on site conditions is a loose approximation at best. Although
bioaccumulation and biomagnification are real concerns, field studies of
past disposal sites have indicated that significant chemical impact on
biological systems is uncommon.

Anderlini et al. (1975) monitored benthic invertebrates for potential
accumulation following the disposal of 10,000 m$^3$ of moderately polluted
contaminants from Oakland Harbor, California. Comparison with the Interim
Plan sediment criteria (NERBC, 1980), indicates that the Oakland Harbor
contained low concentrations of Pb, Zn, and Cu, moderate concentrations of
Hg and Cd and high concentrations of Ni and Cr. Although, in situ benthic
invertebrates such as the bivalves Macoma bathica, Mytilus edulis,

Ischadium demissum, the amphipod Ampelisca milleri and the polychaete Neanthes succinea did not significantly increase tissue trace metal and chlorinated hydrocarbon concentrations, associated laboratory studies did indicate uptake of dissolved Hg by the Macoma, Mytilus and the polychaete, Pectinaria.

Wright (1978) reviewed the environmental impacts associated with disposal in five sites throughout the United States. Three of the sites included accumulation studies although only two of the three were in marine environments: the Duwamish and Eaton's Neck sites.

The Duwamish site in Puget Sound, Washington has gained attention because of the high level of PCB's in its sediment (5.1 ppm, dry weight) from industry in the area as well as from a 1974 spill. PCB tissue levels in two bottom feeders, the English sole, Parophrys vetulus, and the Alaskan pink shrimp, Pandulus borealis, were compared prior to and 9 months after disposal. Levels in the sole after disposal were lower than predisposal levels whereas levels in shrimp showed no change. The PCB concentrations in the water did increase during disposal operations but returned to predisposal level 30 minutes after completion. These levels were associated with organic particulates. Uptake and assimilation of mercury and chromium in the same species were also found to be insignificant (Teeny and Hall, 1977).

Eaton's Neck, in Western LIS, was a disposal site for 75 years until 1974. When dredged material disposal terminated subsequent accumulation tests indicated that there was no apparent difference in metal concentrations between lobsters taken from the dump site and those from a reference

station about 3 miles east of the site. Cadmium levels in the edible part of the lobsters at both sites were below the Food and Drug Administration standard of 0.5 ppm.

The DAMOS program has monitored the uptake of contaminants by *Mytilus edulis* at site G in LIS (NUSC, 1979a; Arimoto and Feng, 1980). Comparison with animal tissue levels at the site and nearby reference station with predisposal levels indicates that the uptake of Nickel (up 1 to 12 times the baseline data, NUSC, 1979a) and PCB's (Arimoto and Feng, 1980) coincided with periods of hightened disposal activities. As disposal subsided the concentration returned to baseline levels. As with PCB's at Duwamish, uptake did occur but was short-term.

Impact studies in the New York Bight, a well known dumping area for a variety of wastes, have indicated varying results. Lee and Jones (1977) measured Hg, Cd, pesticide and PCB levels in the rock crab, *Cancer irroratus* and the demersal fishes, whiting, *Merluccins bilinearis* and ling *Urophycis chuss* sampled from the Mud Dump site in the New York Bight. Despite highly contaminated sediment (by Interim Plan standards, NERBC, 1980), the tissue levels of the contaminants content were appreciably lower than the FDA guidelines for contaminants in fish and shellfish for human consumption. The authors point out that these species are mobile and may have been exposed to the sediment for only a short time. In contrast to the findings of the latter study, a variety of impacts such as increased incidences of disease and mortality, abnormalities in behavior, repro- duction and growth, have been documented and suggested to be related to solid waste disposal in the New York Bight area (Pearce, 1969, 1970;

Mahoney et al., 1973; Sinderman, 1976). This area receives a large amount of industrial, sewage and other solid wastes in addition to dredged material. The relative contribution of dredged material to these impacts is difficult to assess because of the proximity of the various types of dump sites and the mobility of various species. Disposal activities in Long Island Sound have been primarily limited to dredged material. Field studies to date in this region have not indicated the impacts seen in the New York Bight area.

In summary, it appears that despite the potential for bioaccumulation and biomagnification, past field investigations have not indicated an appreciable long-term impact from the disposal of dredged material in the marine environment. However, such results are not conclusive and future on-site long-term studies such as the DAMOS program, will hopefully shed more light on this matter.

E. Chemical Impacts at Candidate Sites

The potential for long-term biological impacts from chemical contam- ination will vary with the nature of the dredged material, the physical characteristics of the dump sites and the biological resources on or around the dump sites. Although numerous studies have shown that contaminant levels from bulk analyses alone are not indicators of potential impacts, these values may be used as part of an initial screening to determine the need for further testing and the most appropriate site for disposal. If a contaminated sediment is deemed acceptable for open-water disposal,

disposal should occur at sites with the least potential to cause harm to the environment. Thus, the physical and biological characteristics of a dump site should be considered in relation to the chemical quality of the dredged sediment.

Biologically conservative assessments of long-term biological impacts at the candidate sites can be made based on the physical characteristics and the biological resources of the candidate sites (Tables 2 and 5, respectively). Table 2 indicates that site C has the highest tidal velocity of all the candidate sites. In view of the fact that the calculated tidal velocity (50-55 cm/sec) would probably erode most capping materials, this site should be reserved for clean materials with moderate amounts of silt. Although the effects of dispersion on surrounding biota are not known, Peddicord et al. (1978) indicate that periodic contaminated suspended sediment would not have an appreciable effect on survival of marine organisms. The major fisheries adjacent to Site C are lobstering south and west of the site and some recreational fishing for tautog, flounder, blackfish and bluefish (Table 5). These species are mobile and will probably move if they are bothered by the resuspension. Further studies are needed before dispersal of silty sediments at this site is carried out.

Sites D and G have borderline tidal velocities which may be capable of resuspending dredged sediments. There is a variety of biological resources at these two sites such that potential impacts might be significant if chemical contaminants are not properly sequestered. Commercial dragging activity at both sites net good catches of a wide variety of bottom feeding finfish. Site G is frequently fished by party boats for flounder, scup and

fluke. Lobster, although limited in this end of the Sound, are caught south and east of site G (Table 5). Shellfish such as the ocean quahog, surf clam and mussel are also taken at these sites. The uptake of Ni and PCB's by mussles at site G, although not of long-term concern, may be an indication of potential problems that could arise from frequent inter-mittent resuspension of unarmored contaminants.

Table B-5
Biological Resources on and around sites.*

| | | Benthic Community | Shellfish | Lobsters | Commercial Finfish | Recreational finfish |
|---|---|---|---|---|---|---|
| A | Bridgeport | low diversity mud species | – | fishery 0.5 miles W | fluke, scup, flounder around and on site | – |
| B | Branford | low diversity mud species | | North fringe of distribution 2 miles S of fishery | – | bluefish, blackfish, shark |
| C | Six Mile Reef | low diversity sand species | may be blue mussels NE of site | fishery S & W adjacent to site | – | tautog flounder, blackfish, bluefish |
| D | Block Island Sound | no data sand species | ocean quahog | | heavily fished for yellowtail flounder, fluke, scup, butterfish, cod, haddock, whiting and hake | – |
| E | Eaton's Neck | moderate diversity mud and sand species | oyster beds 1.5 miles N of site | prime lobster habitat | winter flounder–January windowpane flounder–April Red Hake – June Scup – Fall | – |
| F | New Haven | moderate diversity mud species | – | fishery 0.6 miles S | fluke, scup, flounder, menhaden W&E (removed from site) | bluefish, blackfish, sharks N, W, & E (removed from site) |
| G | New London | high diversity mud species | ocean quahog surf clam mussels rock crab | fishery S&E within area of low distribution | summer & winter flounder during summer; blueback herring in fall | party boats for flounder, scup, fluke, adjacent W&N; tautog E |

*Dames and Moore (1980)

Site E in western Long Island Sound may also be a particularly

sensitive site. In addition to the commercial finfish taken south of the

site, the entire area is one of the most productive lobster habitats in

Long Island Sound (NUSC, 1979a). Oyster beds are found about 1.5 miles

north of the site. Although the potential for resuspension at this site is

relatively low, the area already receives a substantial portion of its

contaminant loading from the East River (Fitzgerald et al. 1974; Bowman,

1977). The combined or synergistic effects of adding further contaminants

on the above populations are unknown. If this site is used, the quality of

dredged material should be carefully considered.

Sites A, B and F may be generally suitable as candidates for future

use. All three sites have low energy characteristics as well as compar-

atively few local biological resources. The erosion of the silt covered

material near Site F may be indicative of the need to sequester contam-

inated sediments.

In view of the limited data at the candidate sites, it is difficult to

make specific predictions concerning potential bioaccumulation, biomagnifi-

cation and associated ramifications at the candidate sites. However,

literature has not indicated that such impacts have been unequivocally

correlated with the disposal of contamined dredged material. Long-term

chemical and biological monitoring at the disposal sites will be continued

to assess the potential for occurrence.

## IV. Summary

It is apparent that the potential long-term impacts associated with open-water disposal of dredged material are difficult to idenfity and assess. Figure 1 represents a simplified diagramatical scheme exhibiting the interrelationships between the physical, chemical and biological impacts.

## A. Physical Impacts and Associated Biological Ramifications

The increased sedimentation associated with open-water disposal may have two direct long-term physical effects. Mounding of the disposed sediment will permanently alter the bottom topography. Secondary impacts such as changes in water circulation patterns, temperatures and salinities occur mainly in shallow water area and therefore would not be significant for any of the proposed candidate sites. Sedimentation may alter the benthic habitat depending on the magnitude of the difference between the dredged sediments and the natural sediments at the dump site. Such habitat alterations would be significant if mud is deposited on sandy bottoms such as those at Sites C and D. Bottom communities are closely related to sediment type so that a change in sediment type would alter the community structure and its productivity. These impacts may be mitigated if the dredged material is similar to the dump site sediment. Continuous long-term disposal may temporarily enhance productivity by keeping the pioneer recolonizing species in dominance but it may impact community stability. The long-term effects of this instability on the ecosystem are not known but may be small since disposal will impact less than 1 percent of the
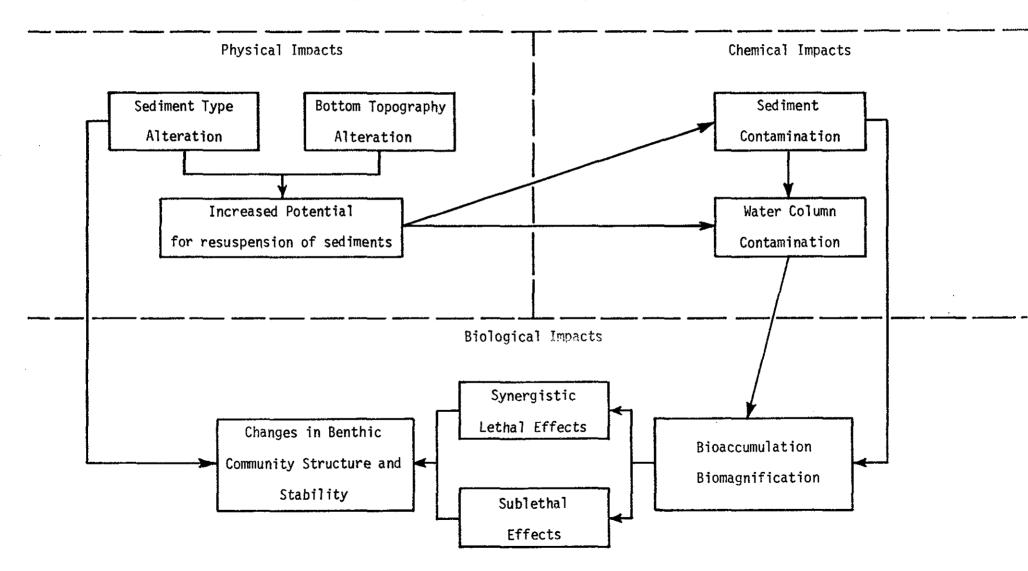
total bottom area of Long Island Sound. Alteration of sediment type and bottom topography may enhance the potential for resuspension of the deposited sediment. Resuspension can further destabilize colonizing benthic organism and/or surrounding populations. Many benthic organisms require a stable substrate. The effects of dispersing silts on a high energy sandy bottom (i.e. Site C) are unknown and need to be studied if this alternative is chosen.

B.  Chemical Impacts and Associated Biological Ramifications.

Many Long Island Sound harbors are "sinks", or receptacles for many river-borne persistent contaminants (heavy metals, petroleum hydrocarbon residues, chlorinated hydrocarbons). Displacement of such contaminants by dredging will introduce these foreign substances to the disposal site sediments. In general, most heavy metals in a reducing environment at a near-neutral pH are sequestered in insoluble geochemical forms. Petroleum and chlorinated hydrocarbons are insoluble in water and are generally tightly bound to organic particulates and clay. Such substances are likely to remain unavailable if the disposal mound remains undisturbed.

Extensive resuspension by bottom currents or biological activity could release sediment contaminants into the water column or change the contaminants into more available forms. Disposal of highly contaminated dredged materials should be reserved for low energy sites and covering with cleaner material should be considered to reduce the possible dispersion of contaminants from the disposal site. Bioaccumulation of contaminants may cause sublethal effects such as reproductive, developmental and growth im-

Figure B-1 Interrelationships between the potential long-term adverse impacts
of open water dredged material disposal on the marine environment*



*Modified from Conner et al (1979).

pairment, deviations in behavior, or cause mutagenic or carcinogenic abnormalities. Synergistic effects may cause death to an individual or population. Persistent contaminants such as DDT could be magnified up the food chain. These effects can contribute to changes in benthic community structure and productivity on or around the disposal site. The relative contribution to such impacts from contaminated dredged material is not known although field studies have not indicated a causative relationship. Properly chosen sites should have important biological resources located far enough away so that the potential for impact on these populations can be minimized. In this regard, Site E is particularly sensitive; careful consideration should be given to disposal of contaminated material at this site given the contaminant loading from the East River.

Although these impacts theoretically can occur, scientific studies to date indicate that contaminated dredged sediments alone are not a causative agent. Studies which have identified problems have not distinguished the impacts from the contaminated dredged material from other forms of contaminant loading. More data are needed before adequate assessment of long-term impacts can be made.

References

Amiard - Triquet, C. 1975. Etude comparative de la contamination par le cobalt 60 et le cesium 137 de quelques invertbres marins fouisseurs. vie et Milieu 25:19-40.

Anderlini, V.C., J.W. Chapman, A.S. Newton, and R.W. Risebrough. 1975.
Dredged disposal study San Francisco Bay & Estuary. Appendix I - Pollutant Availability Study. U.S. Army Corps of Engineers. San Francisco, California.

Arimoto, R., and S.Y. Feny. 1980. Changes in the level of PCB's in Mytilus edulis associated with dredge spoil disposal. DAMOS contribution #10. Marine Sciences Institute, University of Connecticut, Groton, Connecticut, submitted to Corps of Engineers, New England Division, Waltham, MA.

Blumer, M. and J. Sass. 1972. Oil pollution: persistence and degradation of spilled fuel oil. Science 176:1120-1122.

Bokuniewicz, H.J. and R.R. Gordon. 1977. Containment of particular wastes of open-water disposal sites. Pages 109-129 in Ocean Dumping and Marine Pollution, H.D. Palmer and M.G. Gross, eds. Dowden, Hutchinson & Ross, Inc. Stroudsburg, Penn.

Bokuniewicz, H.J., J. Gebert, R.B. Gordon, P. Kaminsky, C.C. Pilbeam, M. Reed, and C. Tuttle. 1977.
Field Study of the effects of storms on the stability and fate of dredged material in subaqueous disposal areas. DMRP Technical Report D-77-22 Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Brannon, J.M., R.M. Engler, J.R. Rose, P. G. Hunt, I. Smith. 1976. Selective analytical partitioning of sediments to evaluate potential mobility of chemical constitutents during dredging and disposal operations. DMRP Technical Report D-76-7, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Brannon, J.M., R.H. Plumb, Jr., and I. Smith. 1978.
Long-term release of contaminants from dredged material. DMRP Technical Report D-78-49, Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Brannon, J.M. 1978. Evaluation of dredged material pollution potential. DMRP Technical Report DS-78-6, Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Bowman, M.J. 1977. Nutrient distributions and transport in Long
    Island Sound. Est. and Coast. Mar. Sci. 5:531-548.

Britt, D. and J.M. Hushon. 1976. Biological effects, criteria and
    standards for hazardous pollutants. Mitre Technical Report
    7283, The Mitre Corporation, McLean, Virginia.

Burke, S.A. and R.M. Engler. 1978. Water quality impacts of
    aquatic dredged material disposal (laboratory investigations).
    DMRP Technical Report DS-78-4, Environmental Laboratory, U.S.
    Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Butman, B. and D. W. Folger. 1978. An instrument system for long
    term sediment transport studies on the Continental Shelf.
    Open file #78-1019, U.S. Geol. Survey, Woods Hole, Mass. 24 pp.

Chase, G.L. 1975. Diving for EIS. Water Spectrum 6(1):27-34.

Chen, K.Y., S.K. Gupta, A.Z. Sycip, J.C. SoLu, M. Knezevic, and W.
    Choi. 1976.
    Research study on the effect of dispersion, settling and
    resedimentation on migration of chemical constitutents during
    open water disposal of dredged materials. DMRP Contract Report
    D-76-1. Environmental Laboratory, U.S. Army Engineer Waterways
    Experiment Station, Vicksburg, Miss.

Cobb, S.P., J.R. Reese, M.A. Granat, B.W. Holliday, E.H. Klehr, and
    J.H. Carroll. 1977.
    Aquatic disposal field investigations, Eaton's Neck Disposal
    Site, Long Island Sound, an environmental inventory. DMRP
    Technical Report D-77-6. Environmental Laboratory, U.S. Army
    Engineer Waterways Experiment Station, Vicksburg, Miss.

Conner, W.G., D. Aurand, M. Leslie, J. Slaughter, A. Amr, and F.I.
    Ravenscroft. 1979.
    Disposal of dredged material within the New York District:
    Volume 1. Present practices and candidate alternatives.
    Prepared by New York District, U.S. Army Corps of Engineers, New
    York, NY

Corps of Engineers, New England Division (CE). 1980. Environmental
    Atlas of New England Channel and Harbor Bottom Sediments.
    Volume I. Federal projects within Long Island Sound and Fishers
    Island Sound.

Dames and Moore. 1980. Environmental Impact Report. Open Water
    Disposal of Dredged Material in Long Island Sound, Block Island
    Sound and Adjacent Waters. Prepared for New England Division,
    Corps of Engineers, Waltham, Mass.

DiSalvo, L.H., H.E. Guard, N.D. Hirsch, and J. Ng. 1977. Assessment and significance of a sediment - associated oil and grease in aquatic environments. DMPR Technical Report D-77-26. Environmental Laboratory, U.S. Army Engineer Waterways Station, Vicksburg, Miss.

Elwood, J.N., S.G. Hildebrand, and J.J. Beauchamp. 1976. Contribution of gut contents to the concentration and body burden of elements in Tipula sp. from a spring-fed stream. J. Fish. Res. Brd. Can. 33:1930-1938.

Engineering Science Inc. 1977. An evaluation of oil and grease contamination associated with dredged material containment areas. DMRP Technical Report D-77-25. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Engler, R.M., J.M. Brannon, J. Rose and G. Bigham. 1974. A practical selective extraction procedure for sediment characterization procedure for sediment characterization. U.S. Army Corps of Engineers, Vicksburg, Miss., paper presented at 168th America Chemical Society National Meeting, Atlantic City, N.J., September 1974.

Environmental Protection Agency. 1976. Quality Criteria for Water. Washington, D.C.

Environmental Protection Agency/Corps of Engineers Technical Committee on Criteria for Dredged and Fill Material. 1977. Ecological evaluation of proposed discharge of dredged material into ocean waters. Environmental Effects Laboratory. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Fitzgerald, W.F., R.J. Szechtman, C.D. Hunt. 1974. A preliminary budget for Cu and Zn in Long Island Sound. Pages 15-42 in investigation on concentrations, distributions, and fates of heavy metal wastes in parts of Long Island Sound. Final report submitted to the Office of Sea Grant Programs, NOAA, Rockville, Maryland.

Federal Register. 1978. Council on Environmental Quality National Environmental Policy Act Implementation of Procedural Provisions; final regulations. 29 November 1978. 43(230):56004

Federal Register. 1979. Environmental Protection Agency Guidelines for Specification of Disposal Sites for Dredged or Fill Material. Tuesday, 18 September 1979 44(182):54234

Fujiki, M. 1973. The transitional condition of Minimata Bay and the neighboring sea polluted by factory waste water containing mercury. In Advances in Water Pollution Research. Proceedings of the 6th International Conference, Jerusalem, June 18-23, 1972, S.H. Jenkins, ed., Pierganon Press.

Fulk, R., D. Gruber, and R. Wullschleger. 1975. Laboratory study of the release of pesticide and PCB materials to the water column during dredging and disposal operations. DMRP Contract Report D-75-6. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Gambrell, R.P., R.A. Khalid, M.G. Verlou, and W.H. Patrick, Jr. 1977. Transformations of heavy metals and plant nutrients in dredged sediments as affected by oxidation reduction potential and pH. Volume II Materials and Methods/Results and Discussion. DMRP Contract Report D-77-4. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Gambrell, R.P., R.A. Khalid, and W.H. Patrick, Jr. 1978. Disposal alternatives for contaminated dredged material as a management tool to minimize adverse environmental effects. DMRP Technical Report DS-78-8. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Greig, R.A., R.N. Reid, and D.R. Wenzloff. 1977. Trace metal concentrations in Sediments for Long Island Sound. Mar. Pollut. Bull. 8:183-187.

Gordon, R.B. and C.C. Pilbeam. 1974. Environmental consequences of dredge spoil disposal in Central Long Island Sound: Geophysical studies contract report prepared by Yale University for United Illuminating, New Haven, Connecticut.

Harrison, F.L., K.M. Wong and R.E. Heft. 1976. Particulates in radionuclide accumulation in oyster *Crassostrea gigas* in the discharge canal of a nuclear power plant. Pages 9-20 in Radioecology and Energy Resources. Ecological Society of America, Special Publication No. 1.

Hjulstrom, F. 1935. Studies of the morphological activity of rivers as illustrated by the River Fyris. Bull. Geol. Inst. Uppsala 25:221-527.

Jarvinen, A.W., M.J. Hoffman, T.W. Thorslund. 1977. Long-term toxic effects of DDT food and water exposure on fathead minnows (*Pinephales promelas*). J. Fish. Res. Brd. Can. 34:2089-2103.

Johnson, R.G. 1971. Animal-Sediment relations in shallow water benthic communities. Mar. Geol. 11:93-104.

Jones, R.A. and F.G. Lee. 1978. Evaluation of the elutriate test as a method of predicting contaminant release during open water disposal of dredged sediments and environmental impact of open water dredged material disposal. Vol. 1 Discussion. DMRP Technical Report D-78-45. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kerfoot, W.B. and S.A. Jacobs. 1974. Cadmium accrual in a combined wastewater treatment aquaculture system. Proc. 1st Annual NSF Trace Containments Conf., Oak Ridge Laboratory, Oak Ridge, Tenn.

Khalid, R.A, R.P. Gambrell, M.G. Verloo and W.H. Patrick, Jr. 1977. Transformations of heavy metals and plant nutrients in dredged sediments as affected by oxidation reduction potential and pH. Volume 1, Literature Review. DMRP Contract Report D-77-4. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Kornreich, M., B. Fuller and F. Dorigan. 1976. Environmental impact of polychlorinated biphenyls. Mitre Technical Report 7006. The Mitre Corporation, McLean, Virginia.

Lee, G.F. and R.H. Plumb, Jr. 1974. Literature review on research study for the development of dredged material disposal criteria. DMRP Contract Report D-74-1. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Lee, G.F. and R.A. Jones. 1977. Assessment of the environmental significance of chemical contaminants present in dredged sediments dumped in the New York Bight. Submitted to U.S. Army Corps of Engineers, New York District (Unpublished).

Lee, G.F., R.A. Jones, F.Y. Saleh, G.M. Mariani, D.H. Homer, J.S. Bulter and P. Bandyopadhyay. 1978. Evaluation of the elutriate test as a method of predicting contaminant release during open water disposal of dredged sediments and environmental impact of open water dredged material disposal. Vol. II, Data Report. DMRP Technical Report D-78-45. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Lowman, F.G., T.R. Rice, and F.A. Richards. 1971. Accumulation and redistribution of radionuclides by marine organisms. Pages 161-199 in Radioactivity in the Marine Environment. Washington, D.C. National Academy of Sciences.

Luoma, S.N. and E.A. Jenne. 1977. The availability of sediment-
    bound cobalt, silver and zinc to a deposit feeding clam. Pages
    213-230 in Biological Implications of Metals in the Environ-
    ment. 15th Life Sciences Symposium, Hanford, Washington.
    September 29 - October 1, 1975. Engineering Research at
    Development Administration.

Macek, K.J., S.R. Petrocelli and B.H. Sleight, III. 1979.
    Considerations in assessing the potential for, and significance
    of biomagnification of chemical residues in aquatic flood
    chains. Pages 251-268 in Aquatic Toxicology, L.L. Marking and
    R.A. Kimerle, eds., Am. Soc. Test. & Mat., Philadelphia, Penn.

Mahoney, J.B.,. F.H. Midlige, and D.G. Deuel. 1973. A fin rot
    disease of marine and euryhaline fishes in the New York Bight.
    Trans. Amer. Fish. Soc. 102:596-605.

Maurer, D.L., R.T. Keck, J.C. Tinsman, W.H. Leathem, C.A. Wethe, M.
    Huntzinger, C. Lord, T.M. Church. 1978.
    Vertical migration of benthos in simulated dredged material
    overburdens. Vol. 1: Marine Benthos. DMRP Technical Report D-
    78-35. Environmental Laboratory, U.S. Army Engineer Waterways
    Experiment Station, Vicksburg, Miss.

McCall, P.L. 1978. Community patterns and adoptive strategies of
    infaunal benthos of Long Island Sound. J. Mar. Res. 35:221-266.

McNulty, J.K., R.C. Work, and H.B. Moore. 1962.
    Some relationships between the infauna of the level bottom and
    the sediment in South Florida. Bull. Mar. Sci. Gulf Carib.
    12:322-332.

Morton, R.W. and M.C. Miller. 1980. Stamford-New Haven Disposal
    Operation Monitoring Survey Report #7. Unpublished mimeo
    submitted to New England Division, U.S. Army Corps of Engineers.

Navy Underwater Systems Center. 1979a. (DAMOS) Disposal Area
    Monitoring System Annual Data Report - 1978. Supplement F, New
    London Disposal Site. New England Division Corps of Engineers,
    Waltham, Mass.

Navy Underwater Systems Center. 1979b. (DAMOS) Disposal Area
    Monitoring System Annual Data Report - 1978. Supplement I,
    Western Long Island Sound Disposal Sites. New England Division,
    Corps of Engineers, Waltham, Mass.

Neff, J.W., R.S. Foster, J. Frank Slowey. 1978. Availability of
    Sediment - absorbed heavy metals to benthos with particular
    emphasis on deposit-feeding infauna. DMRP Technical Report D-
    78-42. Environmental Laboratory, U.S. Army Engineer Waterways
    Experiment Station, Vicksburg, Miss.

New England River Basins Commission (NERBC). 1980. Interim plan
and environmental impact statement for diposal of dredged
material in Long Island Sound.

Ocean Disposal Conference. 1971. Proceedings, Ocean Disposal
Conference, consponsored by Woods Hole Oceanographic Institution
and New England Division, Army Corps of Engineers at the Marine
Biological Laboratory, Woods Hole, Mass., February 23, 1971.

Pearce, J.B. 1969. The effects of waste disposal in New York
Bight. Sandy Hook Marine Laboratory, U.S. Bureau of Sport
Fisheries and Wildlife, Highlands, New Jersey.

Pearce, J.B. 1970. The effects of solid waste disposal on benthic
communities in the New York Bight. FAD Technology Conference on
Marine Pollution and its Effects on Living Resources and
Fishing. Food and Agricultural Organization of the United
Nations, Rome, Italy.

Peddicord, R.K. and V.A. McFarland. 1978. Effects of suspended
dredged material on aquatic animals. DMRP Technical Report D-
78-29. Environmental Laboratory, U.S. Army Engineer Waterways
Experiment Station, Vicksburg, Miss.

Rhoads, D.C. 1973. The influence of disposal feeding benthos on
water tubidity and nutrient recycling. Am. J. Sci. 273:1-22.

Rhoads, D.C. 1974. Organism - sediment relations on the muddy
seafloor. Oceangr. Mar. Biol. Ann. Rev. 12:263-300.

Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and
production on the estuarine seafloor. Am. Sci. 66:577-586.

Saila, S.B., S.D. Pratt, and T.T. Polger. 1972. Dredged spoil
disposal in Rhode Island Sound. University of Rhode Island,
Kingston, R.I.

Saila, S.B. 1973. Testimony at Third Ocean Disposal Conference, 12
April 1973, sponsored by the Maine Department of Sea and Shore
Fisheries and U.S. Army Corps of Engineers, New England
Division.

Sanders, H.L. 1956. Oceanography of Long Island Sound. X. The
biology of marine bottom communities. Bull. Bing. Oceanogr.
Coll. 15:354-414.

Sanders, H.L. 1958. Benthic studies in Buzzards Bay. I. Animal-
Sediment Relationships Limnol. Oceanogr. 3:245-258.

Saucier R.T., C.C. Calhoun, Jr., R.M. Engler, T.R. Patin, and H.K.
Smith. 1978.
Executive Overview and Detailed Summary DMRP Technical Report
DS-78-22. Environmental Laboratory, U.S. Army Engineer
Waterways Experiment Station, Vicksburg, MS.

Schubel, J.R. 1972. Physical and chemical conditions of Chesapeake
Bay. J. Wash. Acad. Sci. 62:56-87.

Science Applications, Inc. (SAI) 1980. Disposal area monitoring
systems progress report March 15-May 15, 1980, DAMOS
contribution #13.

Scura, E.D. and G.H. Theilacker. 1977. The transfer of the
chlorinated hydrocarbon TCB in a laboratory marine food chain.
Marine Biol. 40:317-325.

Serne, R.J. 1977. Geochemical distribution of selected trace
metals in San Francisco Bay sediments. Pages 280-296 in
Biological Implications of Metals in the Environment.
Proceedings of the 15th Annual Hanford Life Sciences Symposium
at Richland, Washington. 29 September - 1 October 1975.
Technical Information Center, Energy Research and Development
Administration.

Shelton, T.B. and J.V. Hunter. 1974. Aerobic decomposition of oil
pollutants in sediments. J. Water Pollut. Control Fed.
46(9):2256-2270.

Sindermann, C.J. 1976. Effects of coastal pollution on fish and
fisheries with particular reference to the Middle Atlantic
Bight. Am. Soc. Limnol. Oceanogr. Spec. Symp. 2:281-301

Stern, E.M. and W.B. Stickle. 1978. Effects of turbidity and
suspended material in aquatic environments; Literature review.
DMRP Technical Report D-78-21. Environmental Laboratory, U.S.
Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Sternberg, R.W. 1972. Predicting initial motion and bedload
transport of sediment particles in the shallow marine
environment. Chapter 3 in Shelf Sediment and Transport Process
and Pattern. D.J.P. Swift, D.B. Duane and O.H. Pilker, eds.
powden, Hutchinson & Ross, Inc., Stroudsburg, Penn.

Stewart, L. 1978. New London disposal site investigation
chronological record of biological and physical benthic
conditions. In A Compilation of Progress Reports, C.L. Brown,
ed. Naval Underwater Systems Center, New London, Conn.

Teeny, F.M. and A.S. Hall. 1977. Aquatic Disposal Field Investigations, Duwamish Waterway Disposal Site, Puget Sound, Washington; Appendix C: Effects of dreged material disposal on concentration of mercury and chromium in several species of marine animals. DMRP Techncial Report 7724. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Thorson, G. 1957. Bottom communities. Pages 461-534 in Treatise on Marine Ecology and Paleoecology. Vol. Ecology, J.W. Hedgepeth, ed. Mem. Geol. Soc. Am. 67.

Tramontano, J. 1978. A compilation of progress reports: Dredging and Dumping of Spoils, Thames River, Conn. Marine Sciences Institute Quarterly Report for January 1, 1978 Naval Underwater Systems Center: New London, Conn.

Ueda, T., R. Nakamura, and Y. Suzuki. 1976. Comparison of Cd 115 accumulation from sediments and sea water by polychaete worms. Bull. Jap. Soc. Sci. Fish. 42:299-306.

Ueda, T., R. Nakamura, and Y. Suzuki. 1977. Comparison of influences of sediments and sea water on accumulation of radionuclides by worms. J. Radiation Res. 18:84-92.

U.S. Navy. 1979. Draft Environmental Impact Statement, Trident Dredging, Thames River Channel, Groton and New London, New London County, Conn. 177 pp & App. A-H.

Waldichuk, M. 1974. Some biological concerns in heavy metals pollution. Pages 1-57 in pollution and physiology of marine organisms, F.J. Vernberg and W.B. Vernberg, ed., Academic Press, N.Y.

Wolfe, D.A. and T.R. Rice. 1972. Cycling of elements in estuaries. Fisheries Bull. NOAA 70:959-973.

Wright, T.D. 1978. Aquatic dredged material disposal impacts. DMRP Technical Report DS-78-1. Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Young, M.L. 1975. The transfer of Zn 65 and Fe 59 along a Fucus serratus (L.) Littorina obtusata (L.) food chain. J. Mar. Biol. Ass. U.K. 55:583-610.